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Land Use, Stream Habitat and Benthic Invertebrate Assemblages in a Highly Altered Iowa Watershed

Abstract

Knowledge of relationships between land cover (*i.e.*, land use) and abiotic and biotic features of headwater streams enhances our ability to predict and effectively assess conditions in a variety of aquatic ecosystems. We evaluated land use effects on stream condition in an Iowa watershed dominated by intensive row crop agriculture and low- intensity urban development by quantifying relationships among land cover, stream invertebrate assemblages and other stream biophysical characteristics (*i.e.*, invertebrate habitat) at 29 sites. On average, 81% of subbasin land cover was agricultural and 6% of land cover was urban across study sites. High nitrate concentrations (range = 5.6–29.0 mg/L) and high relative abundance of oligochaetes and chironomid midges reflected degraded conditions at all sites. However, agriculture and urban land use appeared to have different effects on stream features. Nitrate concentrations were positively related to agricultural land cover, and turbidity and nitrate concentrations were negatively related to urban land cover ($P \leq 0.05$). Invertebrate densities and taxonomic diversity (*i.e.*, total taxa richness, % EPT) were also positively related to agricultural land cover and negatively related to urban land cover. Regardless of land use, highest invertebrate abundance and taxonomic diversity occurred at sites with abundant coarse particulate organic matter, plants and coarse inorganic substrate. Relationships between land cover and invertebrate variables were strong at both local and subbasin measurement scales. Based on invertebrate assemblages, which integrate multiple instream features, we conclude that urban land use had greater adverse effect on stream condition than agriculture in our study watershed. Although impacts of urbanization on stream invertebrates frequently exceed effects of agriculture, this has not previously been demonstrated in Iowa or other Midwestern landscapes so heavily dominated by agriculture.

Keywords

headwater streams, agricultural land cover, urban land cover, invertebrate assemblages

Disciplines

Entomology | Environmental Indicators and Impact Assessment | Natural Resources Management and Policy | Terrestrial and Aquatic Ecology

Comments

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ABSTRACT.—Knowledge of relationships between land cover (*i.e.*, land use) and abiotic and biotic features of headwater streams enhances our ability to predict and effectively assess conditions in a variety of aquatic ecosystems. We evaluated land use effects on stream condition in an Iowa watershed dominated by intensive row crop agriculture and low-intensity urban development by quantifying relationships among land cover, stream invertebrate assemblages and other stream biophysical characteristics (*i.e.*, invertebrate habitat) at 29 sites. On average, 81% of subbasin land cover was agricultural and 6% of land cover was urban across study sites. High nitrate concentrations (range = 5.6–29.0 mg/L) and high relative abundance of oligochaetes and chironomid midges reflected degraded conditions at all sites. However, agriculture and urban land use appeared to have different effects on stream features. Nitrate concentrations were positively related to agricultural land cover, and turbidity and nitrate concentrations were negatively related to urban land cover ($P \leq 0.05$). Invertebrate densities and taxonomic diversity (*i.e.*, total taxa richness, % EPT) were also positively related to agricultural land cover and negatively related to urban land cover. Regardless of land use, highest invertebrate abundance and taxonomic diversity occurred at sites with abundant coarse particulate organic matter, plants and coarse inorganic substrate. Relationships between land cover and invertebrate variables were strong at both local and subbasin measurement scales. Based on invertebrate assemblages, which integrate multiple instream features, we conclude that urban land use had greater adverse effect on stream condition than agriculture in our study watershed. Although impacts of urbanization on stream invertebrates frequently exceed effects of agriculture, this has not previously been demonstrated in Iowa or other Midwestern landscapes so heavily dominated by agriculture.

INTRODUCTION

Biophysical features of headwater streams are tightly coupled with the surrounding terrestrial landscape, and are therefore useful tools for evaluating human land use impacts on aquatic ecosystem condition (*i.e.*, health, integrity; Allan, 2004; Clarke *et al.*, 2008). Landscape change and other activities associated with agriculture and urban development can adversely affect headwater stream condition in multiple ways. Loss of natural vegetation, construction of artificial drainage systems, increased impervious surface area, application of fertilizers and biocides and discharge of human and animal waste contribute to altered stream hydrology and contaminant inputs to streams (Allan, 2004; Walsh *et al.*, 2005). Increased fluctuations in water flow velocity and discharge (*i.e.*, increased flashiness) can alter stream depth and channel width, and sediment input results in higher turbidity and reduced benthic substrate complexity (Allan, 2004; Walsh *et al.*, 2005). A variety of

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contaminants cause pH shifts, and high nutrient concentrations promote increased biological production and oxygen demand, and ultimately, low dissolved oxygen concentrations (Walsh *et al.*, 2005; Weijters *et al.*, 2009).

Whereas it is generally agreed that land cover (*i.e.*, land use) is an excellent predictor of human impacts on stream condition, the apparent magnitude of effect on stream condition can depend on spatial scale of measurement (Walsh *et al.*, 2007; Weijters *et al.*, 2009). Furthermore, the appropriate scale of land cover measurement must be identified and used for ecological condition assessment and restoration practices to be successful (Walsh, 2004). Although some investigators have found significant relationships between local (*e.g.*, riparian) land cover and stream condition indicators, results from other studies suggest that land cover measured at the watershed or subbasin scale is the better predictor of stream condition (Walsh, 2004; Walsh *et al.*, 2007). Riparian vegetation can mitigate effects of agricultural or urban land use occurring elsewhere in the watershed by reducing sediment and nutrient inputs to streams, reducing solar irradiance and maintaining lower water temperatures, and contributing organic matter that provides invertebrates and fish with habitat and food (Allan, 2004; Walsh *et al.*, 2007). However, artificial drainage systems enable contaminants to bypass riparian buffers in many watersheds (Walsh *et al.*, 2005, 2007).

Watersheds of Iowa, U.S.A., are among the most altered landscapes in the world. Approximately 92% of Iowa's land area is used for production of row crops and livestock, and pressure to increase row crop production is intensifying with elevated demand for biofuels (Secchi *et al.*, 2008; ISU Extension, 2009). Although constituting a comparatively small percentage of area, urban land cover has increased as the number of people residing in Iowa's urban centers has increased (*e.g.,* from 1.26 million in 1950 to 1.71 million in 2009; IDNR, 2000; USCB, 2010). Due to intensive land use and associated contaminant inputs to aquatic ecosystems, 77% of monitored Iowa river and stream segments and 69% of Iowa lakes are classified as ''impaired'' or ''potentially impaired,'' and pollutant loads from Iowa contribute to hypoxia in the Gulf of Mexico (Schilling and Spooner, 2006; IDNR, 2010). With urban expansion and greater economic incentives to intensify agricultural production, threats to aquatic ecosystem condition in Iowa are expected to increase (Secchi *et al.*, 2008; Rayburn and Schulte, 2009).

Knowledge of agricultural and urban impacts on headwater streams, and relevant scales of land cover measurement to detect these impacts, is essential to protecting and restoring aquatic ecosystem condition in Iowa and other Midwestern states. In this study, we quantified relationships among land use and biophysical characteristics of headwater streams in a highly altered Iowa watershed. Our overall objective was to assess agricultural and urban land use effects on stream condition through relationships among land cover, stream benthic macroinvertebrate (hereafter invertebrate) assemblages, and invertebrate habitat characteristics. Specific objectives were to (1) quantify relationships between invertebrate variables, other stream biophysical variables and land cover, (2) evaluate differential and relative effects of agriculture and low- intensity urban development on stream condition and (3) evaluate influence of land cover measurement scale on relationships between land cover and instream features.

Invertebrate assemblage characteristics were a primary focus in this study because of their demonstrated value in headwater stream condition assessment (Brown *et al.*, 2009; Purcell *et al.*, 2009). These assemblages are affected by and therefore reflect a multitude of physical, chemical and biological stream features, including independent and interactive effects of variables that are difficult to measure and detect (Brown *et al.*, 2009; Purcell *et al.*, 2009).

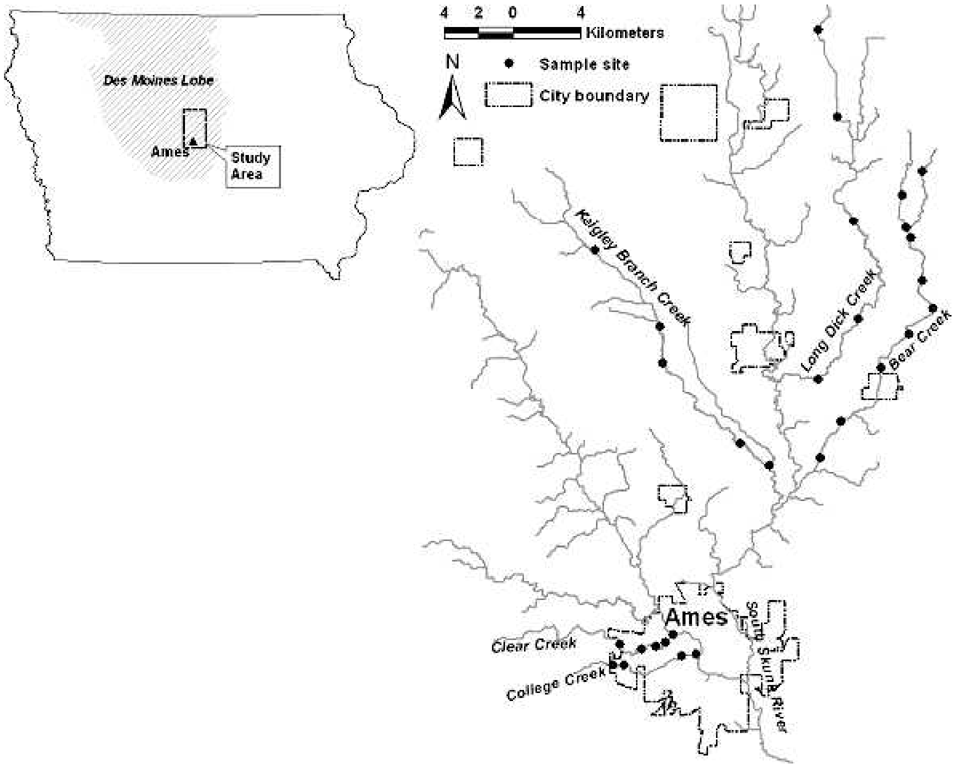

FIG. 1.—Locations of study area, study streams and sample sites

Additionally, invertebrates have critical functional roles in headwater stream ecosystems as predators, consumers of primary producers, processors of organic detritus and as prey (Chadwick *et al.*, 2006; Weijters *et al.*, 2009). Therefore, assemblage characteristics are indicative of biological production, nutrient and energy flow pathways and efficiency, and other functional qualities of streams that are frequently degraded by human land use (Chadwick *et al.*, 2006; Weijters *et al.*, 2009).

Studies of relationships among agricultural and urban land cover, stream invertebrate assemblage characteristics and other instream features have been conducted elsewhere in the Midwest, including Michigan, Minnesota and Wisconsin (*e.g.*, Richards and Host, 1994; Stewart *et al.*, 2001; Stepenuck *et al.*, 2002; Wang and Kanehl, 2003; Weigel, 2003; Nassauer *et al.*, 2004). However, we are aware of no such investigation having previously occurred in a Midwestern landscape as dominated by intensive agriculture as our Iowa study area.

METHODS

STUDY AREA AND SITE DESCRIPTIONS

Study streams were located in the South Skunk River watershed (total area = 4770 km^2) of the Upper Mississippi River basin in central Iowa (Fig. 1). Land in the study watershed had low relief (average basin slope = 3.2%) and at the time of this investigation, 94% of land cover was classified as agricultural (cropland = 68%, pasture = 22%) or urban (4%; NRCS, 2009). Sample sites (n = 29, range of site coordinates = 42°01′11″N–42°21′18″N and 93°28′06″W–93°42′13″W) were distributed across five headwater streams (first- through third order, based on the Strahler method of stream ordering; Fig. 1). Sites in Bear, Long Dick and Keigley Creeks were located in landscapes dominated by agriculture. Sites in College and Clear Creeks were located in Ames, Iowa, a mid-sized urban center with a population of 50,700. However, upstream areas of College and Clear Creeks are located in agricultural areas. Each sample site consisted of a 10 m long section of stream. We attempted to establish sample sites at regularly-spaced points along each stream, yet also select sites that enabled quantification of invertebrate assemblage characteristics across a large land cover and stream habitat gradient. Site selection was also influenced by accessibility (*i.e.*, occurrence of nearby road, landowner permission to access stream). Study sites were located at least 60 m from the nearest road crossing.

LAND COVER

Land cover was measured from the downstream end of each sample site and was quantified at four spatial scales (Morley and Karr, 2002; Walsh *et al.*, 2007), including local (riparian zone within 100 m of the stream, extending 100 m upstream from sample site), 1 k (area within 200 m of the stream, extending 1 km upstream), network (area within 200 m of the stream, for entire upstream length) and subbasin (entire watershed area upstream of sample site; range = 5.7–95.8 km^2 across all sites). At each scale, land cover was quantified as % agriculture (crop land, pasture, gravel surfaces, sewage lagoons), % urban (impervious surfaces, mown lawns) and % natural (ungrazed grassland, trees, water). At each sample site and measurement scale, agricultural, urban and natural cover summed to 100%. ArcMap 9.2 (ESRI, Redlands, CA) was used to delineate and quantify land cover at each spatial scale. Land cover data were obtained from the Iowa Geological Survey and Iowa Department of Natural Resources (IGS and IDNR, respectively) GIS Library, and were digitized and classified at a 15 m resolution. Land cover data within 200 m of the stream were obtained from 2007 USDA National Aerial Photography Program aerial images, and remaining land cover data were acquired from 2002–2003 aerial images. Accuracy of data for area within 200 m of the stream was assessed during field visits, and when necessary, appropriate revisions were made to the land cover data set.

STREAM HABITAT

Measured stream habitat features consisted of biophysical variables that are known to reflect stream condition and affect benthic invertebrate abundance and diversity (Allan, 2004; Walsh *et al.*, 2005). To characterize invertebrate habitat at each site, nitrate-nitrogen (NO_3-N; hereafter nitrate), total phosphorus (hereafter phosphorus), turbidity, pH, temperature, dissolved oxygen and discharge were measured at the downstream end of each sample site during daylight hours on five dates from 14 Jun.–18 Jul. 2007. Each variable was measured every 3–8 d, and except for discharge, measurements were taken in mid channel. Water column concentrations of nitrate and phosphorus were measured from grab samples that were analyzed using standard methods (353.2 for nitrate, 365.1 and 365.3 for phosphorus; USEPA, 1978, 1993a, b). Turbidity, pH, temperature and dissolved oxygen were measured with electronic meters (Hach 2100Q turbidimeter and HQ40d dissolved oxygen/pH/temperature meter), except on two dates when dissolved oxygen was measured using the Winkler method due to electronic probe malfunction. Discharge was measured using methods of Rantz (1982). The stream channel was divided into five cells of equal width, depth was measured at the midpoint of each cell and a current meter (Swoffer 2100 current velocity meter) was used to record four flow velocity readings at 0.6 depth in each cell. Average velocity from these four readings was multiplied by cell cross sectional area, and resulting values (one per cell) were summed to obtain total discharge. For each variable described above, an overall mean for each site was calculated from the five daily measurements taken at that site, and this mean value was used in statistical analyses.

Additional habitat variables were measured on one date, from 9–18 Jul. 2007. Three 0.09-m^2 plot locations were randomly selected from a grid of 100 points that covered the mid channel region of each sample site (*i.e.*, entire stream exclusive of the 0.5 m area adjacent to each bank). A tape measure was used to measure wetted channel width at each sampling plot. A meter stick and current meter were used to measure water depth and flow velocity in the center of each plot. Overhead canopy cover was measured with a spherical densiometer. Inorganic substrate composition within each plot was measured using two methods. First, percent of streambed surface covered by fine ($\leq$2 mm diameter; silt/clay, sand) and coarse ($>$2 mm diameter; gravel, cobble) inorganic material was visually assessed (Gordon *et al.*, 2004). Plot values for fine and coarse substrate abundance summed to 100%. Additionally, a tape measure was extended across the plot at three points (center of plot and two points midway between the center and upstream and downstream plot margins), and streambed contours were followed (Stewart and Garcia, 2002). Ratio of actual distance across the plot, including contours, to the 30 cm horizontal distance across the plot provided a measure of benthic substrate complexity. Measurement of habitat features in plots was performed very carefully to minimize benthic substrate disturbance and prevent dislodgement of benthic invertebrates inhabiting the plot. No invertebrates were observed drifting out of plots while habitat variables were measured.

INVERTEBRATES, CPOM/PLANTS, WOOD DEBRIS

Immediately after measuring physicochemical variables within 0.09 m^2 plots, invertebrates were sampled from plots using methods modified from USEPA's wadeable streams bioassessment protocol (Barbour *et al.*, 1999) and Litvan *et al.* (2008). A D-frame dip net (500 μm mesh) was used to sample these plots for invertebrates and living and nonliving plant material. With the net opening facing upstream and the frame firmly against the substrate at the downstream end of the sample plot, the top 5 cm of substrate was transferred to the net using a shovel and hands. Prior to transferring substrate, cobble and wood debris with less than 50% of surface area contained within the plot were removed and excluded from the sample. After transferring retained material to a bucket, large gravel and cobble particles were scrubbed to remove attached invertebrates and plant material, and then discarded. Remaining material from all three plots at a sample site was combined, and the composite benthic sample material (sand/small gravel, invertebrates, plant material) was preserved in 10% buffered formalin containing rose bengal dye. Formalin was replaced with 70% ethanol within 24 h.

Benthic samples were processed in the laboratory. First, sample contents were emptied into a pan and a comprehensive large-bodied invertebrate search was conducted (all organisms visible to the unaided eye; longest axis $>$0.5 cm) to account for taxa whose densities are often under- or overestimated by subsampling (King and Richardson, 2002). After all large bodied invertebrates were removed, sample contents were homogenized in a pan that was divided into forty-two 38.5 cm^2 cells. A cell was randomly selected, its contents were examined under 10$\times$ magnification, and invertebrates of any size were removed (King and Richardson, 2002). Additional cells were randomly selected and processed in their entirety until at least three cells were processed and $\geq$100 invertebrates were collected (excluding numbers collected during the large-bodied invertebrate search).

Standard keys (Thorp and Covich, 2001; Merritt *et al.*, 2008) were used to identify insects and mollusks to family, whereas most other invertebrate groups were identified to phylum, class or order. Previous investigations demonstrated that the level of taxonomic resolution used in our study is sufficient for detecting ecologically meaningful relationships among environmental and invertebrate community variables (Bailey *et al.*, 2001; Chessman *et al.*, 2007). Invertebrate assemblage variables for each sample site were based on invertebrates collected from sub sampling and the comprehensive large bodied search, accounting for percent of sample processed by sub sampling. Total invertebrate biomass (g AFDW/m^2) was determined by drying invertebrates at 60 C for 24 h and ashing at 500 C for 4 h (APHA, 2005). Density variables (number of individuals/m^2) included EPT density (*i.e.*, total density of Ephemeroptera, Plecoptera, Trichoptera) and individual taxa densities. Taxonomic diversity variables included total taxa richness (number of taxa recorded at each site; taxa/ 0.27 m^2), EPT taxa richness, percent of total invertebrate density composed of EPT (% EPT), Shannon diversity and evenness.

After processing the invertebrate component of a benthic sample, remaining contents were carefully scanned and all pieces of living and dead plant material ≥3 cm in length were removed. All clumps and smaller pieces of plant material (≥1 cm long) were also removed from cells selected for invertebrate sub sampling. Plant material was separated into two categories: wood debris (≥3 cm in length and 1 cm in diameter) and other material that consisted of smaller wood debris and all nonwoody coarse particulate organic matter, living plants and macroalgae (hereafter CPOM/plant). After drying material (60 C for 24 h), wood debris and CPOM/plant abundance (g dry weight/m^2) were quantified. Percent of sample processed was accounted for when determining CPOM/plant abundance.

DATA ANALYSIS

Correlation analyses were used to quantify relationships among land cover and stream habitat. Multiple linear regression was used to identify stream habitat and land cover variables that were related to invertebrate variables (exclusive of individual taxa densities). In regression analysis, all-subsets variable selection (Rathert *et al.*, 1999; Ramsey and Schafer, 2002) was first used to select the set of stream habitat variables that best explained variation in each invertebrate assemblage variable. For each invertebrate variable, the best regression model was considered to be the one in which parameter coefficients for all stream habitat variables were significantly different ($P \leq 0.05$) from zero, and that had lowest values for Mallow's Cp statistic and Akaike's Information Criterion (Ramsey and Schafer, 2002). Subsequently, relationships between land cover and invertebrate variables were evaluated while accounting for effects of stream habitat variables. Using a sequential procedure (Alberti *et al.*, 2007), we added one land cover variable to each previously generated regression model, performed a new regression analysis, and repeated this procedure for each land cover variable measured at each spatial scale. If an invertebrate variable was unrelated to all stream habitat variables based on all-subsets variable selection, simple linear regression analyses were used to quantify the relationship between the invertebrate variable and land cover. Parameter coefficients (β), P-values, and change in model adjusted R^2 were used to assess and compare strength of relationship between invertebrate and land cover variables (Ramsey and Schafer, 2002).

Nonmetric multidimensional scaling (NMDS; Bray Curtis dissimilarity distance matrix) was used to describe relationships among invertebrate taxa densities, stream habitat and land cover. Taxa present at fewer than three sample sites were excluded from NMDS. Vectors representing land cover and stream habitat variables were plotted if they were significantly correlated ($P \leq 0.05$) with the ordination configuration by permutation test

TABLE 1.—Land cover across all 29 sample sites, at each spatial scale of measurement (% land cover; mean with range of values in parentheses)

Measurement scale	Land cover variable	Mean (range of values)
Local	Agriculture	36 (0–93)
	Urban	15 (0–100)
	Natural	49 (0–100)
1 k	Agriculture	46 (0–95)
	Urban	16 (0–82)
	Natural	38 (5–77)
Network	Agriculture	68 (0–89)
	Urban	8 (0–46)
	Natural	24 (9–32)
Subbasin	Agriculture	81 (46–92)
	Urban	6 (1–35)
	Natural	12 (6–20)

(n = 1000). Land cover measurement scales producing the strongest relationships between land cover and invertebrate variables in regression analyses were used in NMDS.

To meet assumptions of parametric statistical tests, data were transformed (log(X + 1) or arcsin square root, where appropriate) prior to statistical analyses (Zar, 1999). Regression and correlation analyses were performed using JMP version 7 (SAS Institute, Cary, NC, U.S.A.), and NMDS was done using R version 2.8.1 (R Development Core Team, Vienna, Austria).

RESULTS

LAND COVER, STREAM HABITAT AND INVERTEBRATE ASSEMBLAGES

On average, agriculture was the most abundant form of land cover at the three largest spatial scales of measurement (Table 1). Across sites, agriculture land cover tended to increase from local (mean = 36%) to subbasin scales (mean = 81%; Table 1). Natural land cover, composed primarily of perennial vegetation, was generally abundant in riparian areas (*i.e.*, local scale; mean = 49%) but declined with increasing distance from the stream (Table 1). Urban land cover at local and subbasin scales averaged 15% and 6%, respectively (Table 1). All land cover types were highly variable across sample sites (Table 1). Similarly, the wide range of measured values for several habitat variables (nitrate, turbidity, dissolved oxygen, discharge, channel width, depth, flow velocity, overhead canopy, coarse substrate, wood debris and CPOM/plant) reflected extensive variability in stream habitat across sites (Table 2).

Forty-two invertebrate taxa were recorded in this study (Table 3). Chironomidae (midges) and Oligochaeta (oligochaetes) tended to dominate assemblages, with individuals of these taxa constituting 74% of recorded invertebrates (Table 3). Average densities of turbellarians, nematodes, physid gastropods, sphaeriid bivalves and elmid beetles were also relatively high (Table 3). Comparative rarity of most taxa was reflected in values for total invertebrate taxa richness across sites (mean ± SE = 10.7 ± 0.78 taxa/0.27 m^2). Mean (±SE) values for additional invertebrate metrics were as follows: total invertebrate biomass (0.54 ± 0.13 g AFDW/m^2), EPT density (405 ± 116 individuals/m^2), EPT taxa richness (3.2 ± 0.5 taxa/0.27 m^2) and % EPT (8.8 ± 2.3).

TABLE 2.—Mean values (with range of values in parentheses) for stream habitat variables across all 29 sample sites

Stream habitat variable (measurement units)	Mean (range of values)
Nitrate (mg/L)	14.0 (5.6–29.0)
Total phosphorus (mg/L)	0.08 (0.04–0.14)
Turbidity (NTU)	16.6 (4.0–35.4)
pH	8.3 (7.7–8.5)
Temperature (C)	20.1 (17.2–23.6)
Dissolved oxygen (mg/L)	7.2 (2.4–15.3)
Discharge (m^3/s)	0.268 (0.004–1.058)
Channel width (m)	3.9 (1.0–8.2)
Depth (m)	0.17 (0.05–0.40)
Flow velocity (m/s)	0.13 (0–0.43)
Overhead canopy (% cover)	34 (0–88)
Coarse substrate (% cover)	27 (0–78)
Wood debris (g/m^2)	23 (0–128)
CPOM/plant (g/m^2)	69 (1–450)

RELATIONSHIPS AMONG LAND COVER, STREAM HABITAT AND INVERTEBRATE VARIABLES

Thirteen of 16 stream habitat variables were included in statistical analyses of relationships among land cover, stream habitat and invertebrate assemblages. Discharge was omitted because this variable was strongly related to several other habitat variables (Table 4), and channel width, depth and flow velocity appeared to be adequate proxies for discharge. Correlative relationship strength verified expectations that coarse substrate abundance was an adequate surrogate for substrate complexity (Pearson $r = 0.52$; $P < 0.01$) and the latter was excluded from statistical analyses. Fine inorganic substrate abundance was eliminated from analyses because it was perfectly correlated with coarse substrate abundance.

Correlation analyses revealed significant relationships between land cover and several stream habitat variables. Across measurement scales, stream sites with high agricultural land cover generally had higher nitrate and diurnal dissolved oxygen concentrations, lower pH and overhead canopy cover and greater depths than sites with high urban or natural land cover (Table 5). Additionally, turbidity was negatively related to urban land cover, and CPOM/plant abundance was negatively related to natural land cover (Table 5).

EPT density, total invertebrate taxa richness, Shannon diversity, EPT taxa richness and % EPT were significantly related to stream habitat variables (Table 6). Regression results suggested that turbidity, dissolved oxygen, coarse substrate and CPOM/plant abundance were especially useful predictors of invertebrate abundance and diversity. All five aforementioned invertebrate assemblage variables were positively related to coarse substrate abundance, and all but total taxa richness were positively related to turbidity (Table 6). Additionally, total taxa richness, Shannon diversity and % EPT were positively associated with CPOM/plant abundance, and EPT density and taxa richness and % EPT were positively related to diurnal dissolved oxygen concentration (Table 6).

After accounting for related stream habitat variables, addition of land cover to regression models revealed that three invertebrate assemblage variables were positively related to agriculture and negatively related to urban land cover at one or more spatial scales (Table 7). EPT density was positively related to agriculture and negatively related to urban land cover at the local scale, and the same relationships held for % EPT at the subbasin scale (Table 7). Total invertebrate taxa richness was positively related to agricultural land cover at

TABLE 3.—Density values for invertebrate taxa across all 29 sample sites (mean number of individuals/m^2 with SE in parentheses)

Taxon	Mean density (SE)
Platyhelminthes	
Turbellaria	307 (299)
Nematoda	101 (35)
Mollusca	
Gastropoda	
Physidae	232 (123)
Planorbidae	17 (13)
Bivalvia	
Sphaeriidae	105 (54)
Annelida	
Oligochaeta	1496 (443)
Euhirudinea	10 (7)
Arthropoda	
Acariformes	21 (16)
Insecta	
Ephemeroptera	
Ameletidae	32 (27)
Baetidae	58 (16)
Caenidae	64 (26)
Ephemeridae	18 (8)
Heptageniidae	39 (18)
Isonychiidae	2 (1)
Leptohyphidae	70 (33)
Leptophlebiidae	1 (1)
Siphlonuridae	2 (2)
Odonata	
Coenagrionidae	5 (5)
Gomphidae	<1 (<1)
Libellulidae	<1 (<1)
Plecoptera	
Perlidae	1 (1)
Trichoptera	
Helicopsychidae	9 (5)
Hydropsychidae	61 (27)
Hydroptilidae	42 (17)
Leptoceridae	8 (7)
Limnephilidae	<1 (<1)
Hemiptera	
Corixidae	1 (1)
Coleoptera	
Dytiscidae	13 (6)
Elmidae	153 (63)
Haliplidae	5 (4)
Hydrophilidae	3 (3)
Psephenidae	<1 (<1)

TABLE 3.—Continued

Taxon	Mean density (SE)
Megaloptera	
Corydalidae	<1 (<1)
Sialidae	1 (1)
Diptera	
Ceratopogonidae	29 (12)
Chironomidae	2816 (479)
Empididae	40 (17)
Simuliidae	66 (48)
Tipulidae	8 (3)
Malacostraca	
Amphipoda	5 (3)
Decapoda	1 (1)
Isopoda	
Asellidae	24 (17)

all but the network scale and negatively related to urban cover at local and subbasin scales (Table 7). Finally, natural land cover was negatively related to total invertebrate biomass at local and network scales and negatively related to EPT density, total taxa richness and EPT taxa richness at the subbasin scale (Table 7).

Based on frequency of statistically significant associations, and change in adjusted R^2 after land cover addition to previously constructed regression models, overall relationships between land cover and invertebrate assemblage variables were strongest at local and subbasin scales (Table 7). These two land cover measurement scales were included in NMDS analyses of invertebrate taxa densities (Fig. 2). The simplest NMDS ordination which minimized stress (final stress = 14.95) contained three dimensions; variables related to the first pair of axes in this ordination are displayed (Fig. 2). Similar to patterns for other invertebrate assemblage variables (Table 7), densities of many invertebrate taxa were negatively related to urban cover, including several EPT taxa (*e.g.,* Caenidae, Isonychiidae, Leptohyphidae, Helicopsychidae, Hydroptilidae, Leptoceridae; Fig. 2). At the subbasin scale, these same taxa were also positively associated with agriculture and negatively related to natural land cover (Fig. 2B). Although relatively few taxa (*e.g.,* turbellarians) were positively associated with urban cover, oligochaetes and chironomids, whose densities dominated the overall invertebrate assemblage (Table 3), tended to be most abundant at sites with high urban land use (Fig. 2).

Similar to other invertebrate assemblage variables (Table 6), variation in invertebrate taxa densities was strongly associated with turbidity and CPOM/plant and coarse substrate abundance (Fig. 2). Sites with high CPOM/plant abundance, greater depth and relatively low pH supported relatively high densities of physid gastropods, dytiscid beetles, simuliid flies and asellid isopods and relatively low densities of heptageniid mayflies (Fig. 2). Sites with abundant coarse substrate and high turbidity and temperatures were most prevalent in agricultural areas (Fig. 2B). These locations typically supported highest densities of EPT taxa and elmid beetles (Fig. 2).

DISCUSSION

Across our 29 sample sites, 88–94% of subbasin land area had been altered for agricultural and urban uses, and values for habitat and invertebrate assemblage variables

TABLE 4.—Pearson correlation coefficients (r) for relationships among stream habitat variables. TP = phosphorus, Temp. = temperature, DO = dissolved oxygen, Q = discharge, Width = channel width, Flow = flow velocity, Canopy = overhead canopy, Wood = wood debris. Values in bold indicate statistically significant relationships ($P \leq 0.05$)

Variable	TP	Turbidity	pH	Temp.	DO	Q	Width	Depth	Flow	Canopy	Coarse substrate	Wood	CPOM/plant
Nitrate	0.21	−0.01	−0.33	0.21	**0.46**	0.24	−0.10	0.31	−0.29	**−0.49**	−0.16	**−0.40**	0.07
TP		**0.46**	0.03	**0.49**	0.08	**0.48**	0.33	0.28	−0.06	−0.33	−0.09	0.17	0.13
Turbidity			0.35	**0.58**	−0.14	**0.85**	**0.45**	0.15	0.29	−0.25	0.14	0.09	**−0.43**
pH				**0.41**	**−0.37**	**0.45**	**0.42**	**−0.41**	0.26	0.30	**0.40**	0.24	**−0.52**
Temp.					0.13	**0.66**	**0.60**	0.18	−0.03	−0.27	0.03	−0.10	−0.29
DO						−0.05	−0.04	0.35	−0.03	−0.21	**−0.46**	−0.16	**0.38**
Q							**0.39**	0.08	0.26	−0.32	0.24	−0.05	**−0.51**
Width								0.09	−0.19	0.11	0.03	−0.11	−0.14
Depth									−0.29	**−0.51**	**−0.46**	−0.09	**0.39**
Flow										0.01	0.21	0.01	−0.33
Canopy											0.24	0.27	−0.05
Coarse substrate												0.02	**−0.40**
Wood													0.30

TABLE 5.—Pearson correlation coefficients (r) for relationships between land cover and stream habitat variables, by land cover measurement scale. For land cover scales, NW = Network and Basin = subbasin. For stream habitat variables, TP = total phosphorus, Temp. = temperature, Width = channel width, Flow = flow velocity, DO = dissolved oxygen, Canopy = overhead canopy cover, Wood = wood debris. Values in bold indicate statistically significant relationships ($P \leq 0.05$)

	Agriculture				Urban				Natural			
	Local	1 k	NW	Basin	Local	1 k	NW	Basin	Local	1 k	NW	Basin
Nitrate	**0.58**	**0.64**	**0.91**	**0.74**	**−0.62**	**−0.73**	**−0.75**	**−0.71**	−0.07	−0.20	**−0.66**	**−0.65**
TP	0.23	0.30	0.18	0.24	−0.17	−0.27	−0.12	−0.18	−0.06	−0.21	−0.23	−0.36
Turbidity	0.18	0.24	0.12	0.33	−0.30	**−0.42**	**−0.37**	**−0.39**	0.16	0.15	0.30	−0.23
pH	**−0.43**	**−0.50**	−0.35	**−0.42**	0.14	0.29	0.23	0.34	**0.50**	**0.45**	**0.38**	**0.53**
Temp.	0.14	0.20	0.30	0.22	−0.22	−0.27	−0.23	−0.23	0.12	0.02	−0.28	−0.18
DO	**0.52**	**0.51**	**0.40**	**0.44**	−0.25	**−0.40**	−0.37	**−0.39**	**−0.40**	**−0.43**	**−0.40**	**−0.57**
Width	−0.09	−0.03	−0.02	−0.08	0.00	−0.12	0.05	0.05	0.13	0.18	−0.02	0.13
Depth	**0.42**	**0.50**	**0.44**	**0.44**	−0.14	−0.34	−0.35	**−0.39**	**−0.40**	−0.34	**−0.43**	**−0.50**
Flow	0.05	−0.02	−0.27	−0.08	0.03	0.06	−0.04	0.05	−0.10	−0.12	**0.39**	0.02
Canopy	**−0.80**	**−0.64**	**−0.51**	**−0.66**	**0.56**	**0.58**	**0.57**	**0.62**	**0.48**	**0.40**	0.24	**0.70**
Coarse substrate	−0.27	−0.24	−0.17	−0.20	0.02	0.13	0.12	0.13	**0.40**	0.29	0.27	0.30
Wood	−0.28	−0.24	**−0.37**	−0.30	0.19	0.35	0.29	0.32	0.16	−0.03	0.21	0.17
CPOM/plant	0.15	0.27	0.09	0.07	0.17	0.03	0.08	0.03	**−0.45**	**−0.49**	**−0.43**	−0.27

TABLE 6.—Regression models (best of all possible subsets) of relationships between invertebrate and stream habitat variables

Invertebrate variable	Model Adj. R^2	Partial R	P value	Stream habitat variable
EPT density	0.32	0.45	0.020	Turbidity
		0.49	0.010	Dissolved oxygen
		0.46	0.015	Coarse substrate
Total taxa richness	0.47	−0.40	0.037	Overhead canopy
		0.64	<0.001	Coarse substrate
		0.65	<0.001	CPOM/plant
Shannon diversity	0.42	0.42	0.029	Turbidity
		0.47	0.013	Coarse substrate
		0.69	<0.001	CPOM/plant
EPT taxa richness	0.38	0.48	0.012	Turbidity
		0.45	0.018	Dissolved oxygen
		0.54	0.004	Coarse substrate
% EPT	0.57	0.73	<0.001	Turbidity
		0.43	0.032	Dissolved oxygen
		−0.47	0.018	Depth
		0.51	0.009	Coarse substrate
		0.42	0.039	CPOM/plant

were indicative of degraded stream conditions throughout the study watershed. Nitrate concentrations at study sites generally exceeded summer values recently reported from streams in other row-crop dominated watersheds of the Midwestern U.S.A., where nitrate concentrations are consistently among the highest in the nation (Bernot *et al.*, 2006; Figueroa-Nieves *et al.*, 2006; Schilling and Spooner, 2006; Heatherly *et al.*, 2007; Renwick *et al.*, 2008; Wagner *et al.*, 2008; Diebel and Vander Zanden, 2009; Warrner *et al.*, 2009). Additionally, nitrate concentration at each of our study sites (range = 5.6–29.0 mg/L) was higher than 3.2 mg/L, a threshold value that, based on a USEPA Wadeable Streams Assessment, is indicative of poor conditions in Midwestern streams (Van Sickle and Paulsen, 2008). Turbidity in our study was similar to levels recorded in another study of a Midwestern stream within a row-crop dominated watershed, but on average much higher than turbidity recorded in cattle grazed landscapes of southern Minnesota (Sovell *et al.*, 2000; Figueroa-Nieves *et al.*, 2006). Diurnal oxygen concentrations in both urban and agricultural areas of our study were occasionally below minimum levels considered necessary to support stream biota (5 mg/L; IDNR, 2002). Invertebrate assemblages at nearly all study sites were dominated by oligochaetes and midges, an observation also consistent with degraded stream condition (Walsh *et al.*, 2005; Carlisle *et al.*, 2008).

LAND COVER, STREAM HABITAT AND INVERTEBRATES

Our results suggest that stream condition in our study watershed was affected by land use, and agriculture and urban development had different effects on stream biophysical features. Higher turbidity and nitrate concentrations in agricultural areas suggest that farming practices resulted in greater nutrient and sediment inputs to streams than urban development. However, invertebrate abundance and diversity, including several related variables frequently used to assess ecological condition (*e.g.*, total taxa richness, EPT density, % EPT) were clearly negatively related to urban land cover. Based on invertebrate assemblage variables, which integrate multiple instream features, we conclude that urban

TABLE 7.—Invertebrate variables significantly related to land cover variables after accounting for stream habitat variables (see Table 6). A P value is reported when there was a significant relationship between an invertebrate variable and land cover variable at a specific spatial scale of measurement. β = parameter coefficient for land cover variable. Δ Adj. R^2 = Adj. R^2 from model including land cover minus Adj. R^2 of model with only stream habitat variables

	Local			1 k			Network			Subbasin		
	P value	β	Δ Adj. R^2	P value	β	Δ Adj. R^2	P value	β	Δ Adj. R^2	P value	β	Δ Adj. R^2
Agriculture												
EPT density	0.031	0.75	0.11	ns	—	—	ns	—	—	ns	—	—
Total taxa richness	0.020	0.19	0.09	0.043	0.14	0.07	ns	—	—	0.013	0.46	0.11
% EPT	ns	—	—	ns	—	—	ns	—	—	0.039	0.42	0.06
Urban												
EPT density	0.011	−0.97	0.15	ns	—	—	ns	—	—	ns	—	—
Total taxa richness	0.036	−0.16	0.07	ns	—	—	ns	—	—	0.016	−0.43	0.10
% EPT	ns	—	—	ns	—	—	ns	—	—	0.028	−0.43	0.07
Natural												
Total biomass	0.045	−0.21	NA	ns	—	—	0.006	−1.01	NA	ns	—	—
EPT density	ns	—	—	ns	—	—	ns	—	—	0.006	−6.83	0.17
Total taxa richness	ns	—	—	ns	—	—	ns	—	—	0.008	−1.29	0.12
EPT taxa richness	ns	—	—	ns	—	—	ns	—	—	0.047	−1.61	0.07

land use had greater adverse effect on stream condition than agriculture in our study watershed. At the subbasin scale, urban land cover averaged 6% (range = 1–35% and 0–31%, respectively) of total land cover across our 29 study sites. Similar to our findings, adverse effects of urbanization on invertebrate assemblages in Wisconsin were detected when average urban land cover ranged from 2–12% of watershed land area (Stewart *et al.*, 2001; Stepenuck *et al.*, 2002; Wang and Kanehl, 2003). However, agricultural land cover in our study far exceeded that recorded in these Wisconsin watersheds. Therefore, we provide unique evidence that even under extreme agricultural land use, low-intensity urban development has overwhelming impacts on stream condition.

As in our Iowa study, nitrate concentrations were observed to be greater in predominantly agricultural watersheds than urbanized watersheds in other Midwestern states (Nassauer *et al.*, 2004; Diebel *et al.*, 2009). Fertilizer application, erosion attributed to mechanical tillage and exposed soil, and rapid movement of nutrients and sediment to streams via tile drains were likely primary causes for high nitrate and turbidity at our sites and in other agricultural streams (Nerbonne and Vondracek, 2001; Bernot *et al.*, 2006). In our study, natural vegetation, including riparian tree cover (*i.e.*, natural land cover), was less abundant in agricultural areas than in urban landscapes, potentially contributing to elevated inputs of nutrients and sediment at agricultural sites (Nerbonne and Vondracek, 2001; Schultz *et al.*, 2004). Value of natural vegetation in protecting stream integrity by absorbing surface water and filtering sediment and nutrients from overland flow is well documented, as are beneficial local scale (*i.e.*, riparian) functions of regulating stream water temperature and

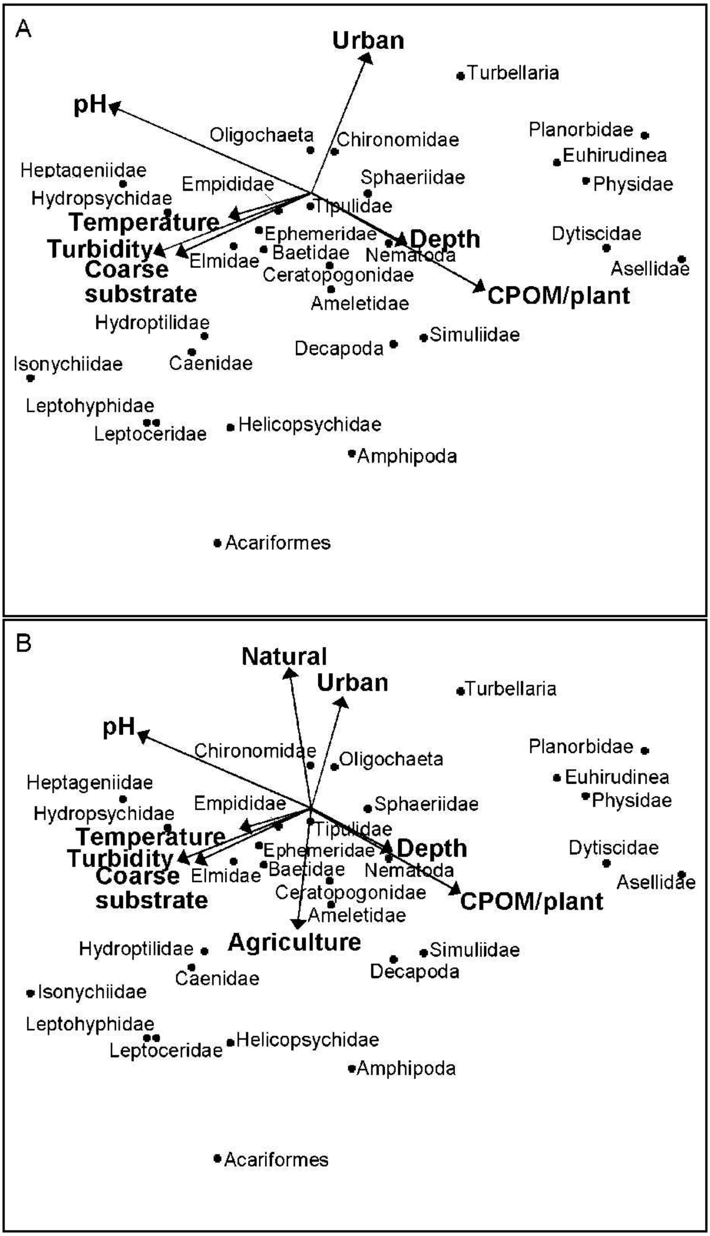

FIG. 2.—Nonmetric multidimensional scaling (NMDS) results from ordination of invertebrate taxa densities. Stream habitat and land cover variables measured at (A) local and (B) subbasin scales are plotted as vectors if they were significantly related to invertebrate densities ($P \leq 0.05$). Vector length and direction reflects strength and direction of relationship between the habitat or land cover variable and invertebrate densities

contributing organic matter that provides food and habitat to stream organisms (Nerbonne and Vondracek, 2001; Allan, 2004). In combination with high nitrate availability, the resulting greater light intensity might have stimulated primary production and caused elevated diurnal dissolved oxygen at our agricultural stream sites (Allan, 2004; Renwick *et al.*, 2008). Field observations revealed that negative relationships between CPOM/plant abundance and natural land cover at relatively small land cover measurement scales were due to greater abundance of macroscopic algae and plants at unshaded agricultural sites. These observations supported hypotheses that dissolved oxygen was influenced by instream primary production.

Negative associations between urban land cover and stream invertebrate taxonomic diversity (*e.g.*, total taxa richness, % EPT) were found in Minnesota and Wisconsin, as they were in our study (Stewart *et al.*, 2001; Stepenuck *et al.*, 2002; Wang and Kanehl, 2003; Weigel, 2003). Causes for adverse urban land use effects on stream invertebrate assemblages were not identified in those studies or in our study. Many investigators have recognized the difficulty of identifying causal linkages among land cover, instream habitat and biotic assemblage structure because it is impossible to account for all environmental factors (*e.g.*, biophysical stressors) which operate independently and in combination to structure assemblages over variable spatial and temporal scales (Paul and Meyer, 2001; Carlisle *et al.*, 2008). In our study, lower dissolved oxygen concentrations at urban stream sites might have contributed to negative relationships between urban land cover and low EPT abundance and diversity (Nerbonne and Vondracek, 2001; Paul and Meyer, 2001). However, even lower nocturnal oxygen concentrations might have occurred at comparatively eutrophic agricultural sites (Allan, 2004; Warrner *et al.*, 2009). Because high sediment delivery to streams adversely affects most invertebrate taxa, positive relationships between turbidity and invertebrate abundance and diversity were contrary to expectations (Nerbonne and Vondracek, 2001; Roy *et al.*, 2003). These relationships were likely statistical artifacts of urban impacts on invertebrates, and better overall conditions in turbid, agricultural streams. Significant relationships between pH and land cover were also difficult to explain and interpret due to multiple chemical constituents that influence pH, and minimal variation in pH across our study sites. Finally, positive effects of natural terrestrial vegetation on stream habitat and invertebrates, frequently observed elsewhere (Nerbonne and Vondracek, 2001; Allan, 2004), were not apparent in our study. If natural land cover had any positive effects on stream invertebrate abundance and diversity in our study watershed, these effects were overwhelmed by adverse effects of land use, especially urban development.

In our study, stream habitat variables most strongly related to invertebrate variables were not significantly related to urban or agricultural land cover. Regardless of land cover, highest invertebrate abundance and diversity occurred at sites with high CPOM/plant and coarse substrate abundance. These findings were consistent with known environmental requirements of aquatic invertebrates. Macroalgae, plants and detritus provide food and habitat for invertebrates, and structurally complex habitat in the form of coarse substrate provides attachment and grazing sites, and refuges from predators and physical disturbance (Hershey and Lamberti, 2001; Hrodey *et al.*, 2009).

LAND COVER MEASUREMENT SCALE

Whereas measurement scale had negligible effect on relationships between land cover and stream habitat variables, relationships between land cover and invertebrate variables were strong at both local and subbasin scales in our study. Negative relationships between urban land cover and invertebrate assemblage variables (and presumably stream condition) at the subbasin scale were likely attributed to impervious surface and artificial drainage systems that effectively link the stream to the entire watershed, and dramatically increase delivery rate and quantity of water and contaminants transported to streams from upland landscapes (Allan, 2004; Walsh *et al.*, 2007). These hydrological alterations have been implicated in the inability of upland and riparian vegetation to absorb surface water, intercept pollutants and improve stream condition in urban watersheds (Roy *et al.*, 2005; Walsh *et al.*, 2005).

Although many studies have identified watershed- or subbasin-scale land cover as a good predictor of stream invertebrate assemblages, relationships between local land cover features and stream invertebrate assemblage characteristics have been less consistent (Lammert and Allan, 1999; Weigel *et al.*, 2000; Nerbonne and Vondracek, 2001; Stewart *et al.*, 2001). This may be attributed, in part, to impervious surfaces and subsurface drainage systems that bypass the riparian zone (Allan, 2004; Walsh *et al.*, 2007). Nevertheless, there is clear evidence that in many landscapes, riparian features can regulate water chemistry, physical habitat and biotic assemblage structure in the adjacent stream (Stewart *et al.*, 2001; Allan, 2004).

CONCLUSIONS

Evidence of degraded stream conditions in both agricultural (*e.g.,* high nitrate concentrations) and urban landscapes (*e.g.,* low invertebrate abundance and taxonomic diversity) of our study demonstrates a need to change land use practices in central Iowa. Economic incentives to urbanize remaining natural land cover or use it for agricultural production are formidable barriers to improving environmental conditions in this geographic region (Secchi *et al.*, 2008; Rayburn and Schulte, 2009). Under the likely scenario of expanding intensive land use, greater implementation of best management practices will likely be needed to mitigate anthropogenic impacts on stream condition. Practices potentially benefitting Iowa streams include conservation tillage, wetland construction, planting of riparian vegetation and elimination of direct hydraulic connections between stream channels and impervious surface and artificial drainage systems (Allan, 2004; Walsh *et al.*, 2005, 2007).

Several previous studies investigated relationships among land cover, stream habitat and invertebrate assemblages in the Midwest (*e.g.,* Richards and Host, 1994; Lammert and Allan, 1999; Sovell *et al.*, 2000; Weigel *et al.*, 2000; Nerbonne and Vondracek, 2001; Weigel, 2003; Nassauer *et al.*, 2004; Hrodey *et al.*, 2009), and a few such studies compared agricultural and urban land use effects on invertebrate habitat and assemblages (Stewart *et al.*, 2001; Stepenuck *et al.*, 2002; Wang and Kanehl, 2003). Results from aforementioned studies and our study suggest that urbanization has greater adverse effects than agriculture on invertebrate assemblages in the Midwest. However, our study appears to provide the best evidence that even under extreme agricultural land use, low-intensity urban development can have greater negative impacts on stream condition.

To improve stream condition in central Iowa, our results suggest that land use management should focus at both watershed and local (*i.e.*, riparian) scales and that relationships among land use and benthic invertebrate assemblage characteristics should be used to evaluate human impacts. Regardless of landscape context, invertebrate abundance and diversity in our study were positively associated with high coarse substrate availability and CPOM/plant abundance. These findings suggest that even in highly altered watersheds and degraded streams, such habitat features enhance biological production and diversity and related stream ecosystem functions and services. However, such findings also underscore the importance of land management practices that prevent sediment loading and other causes of reduced coarse substrate and CPOM/plant availability (Paul and Meyer, 2001; Allan, 2004).

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